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Short-term effects of a wildfire on the endangered Dupont's Lark *Chersophilus duponti* in an arid shrub-steppe of central Spain

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Abstract. In Europe, Dupont's Lark *Chersophilus duponti* is a threatened open-habitat bird. Prescribed burning has sometimes been proposed for its conservation, but without evidence of its effectiveness. To evaluate the short-term effects of a summer wildfire on this species, we performed several transect counts in the burnt and unburnt parts of a shrub-steppe in central Spain. The same transects were counted within a three-year interval prior to the fire and were repeated during the first two springs after the fire. We also measured the vegetation during the first two springs after the fire. In the burnt area, we observed a decrease of about 85–100% in Dupont's Lark abundance, and about 7–15% in the control area. The disappearance of the scrub cover after fire and its slow regeneration, as well as the large increase in grass cover during the second year, may explain the decrease in this habitat-specialist bird species. Fire should be avoided in areas occupied by the Dupont's Lark, as its negative effects in the short-term may cause local extinctions. However, prescribed burning may be used in neighboring areas to create new open habitats that may be subsequently colonized by this species.

Key words: conservation tool, disturbance, *Genista pumila*, fire, Mediterranean, páramo, shrub-steppe habitat

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INTRODUCTION

Wildfires belong to the most important phenomena in landscape dynamics and play an important role in determining the distribution of birds typical of forests (Moreira et al. 2003, Jacquet & Prodon 2009, Nappi & Drapeau 2009) and open-habitats (Dechant et al. 2003, Pons & Bas 2005, Bouwman & Hoffman 2007, Menz et al. 2009). Wildfires may provoke opposing changes in the distribution and biology of avian species, such as higher reproductive success (Nappi & Drapeau 2009), lower abundance (Moreira et al. 2003), or higher abundance and species richness (Bouwman & Hoffmann 2007). Some studies have determined that wildfires may have positive effects on the conservation and persistence of some threatened Mediterranean avifauna, since open habitats created in forests by wildfires are often occupied by more specialized and threatened species (Moreira et al. 2001, Pons & Bas 2005, Brotons et al. 2008, Clavero et al. 2011) and habitat quality may improve compared to pre-fire situations after vegetation recovery (Real 2000, Menz et al. 2009).

In the Mediterranean region the effect of fire on avifauna has been studied in some detail in forests and tall scrubland habitats (Herrando & Brotons 2002, Herrando et al. 2002, Clavero et al. 2011) but to our knowledge the effects of fire on open-habitat birds in Mediterranean shrub-steppes has not been analyzed and research on the consequences of fire for plants is very scarce. Likewise, the effect of fire on the distribution and population trends of one of the most endangered steppe passerines in Europe, the Dupont's Lark *Chersophilus duponti*, is totally unknown, both in the short- and long-term.

In Europe, Dupont's Lark is restricted to low scrub areas in Spain (Garza & Suárez 1990, Seoane et al. 2006), where its populations have been reduced by 50% over the last 20 years to only 2,200 pairs and several populations have become extinct (Tella et al. 2005, Suárez 2010). Agricultural intensification, habitat loss, fragmentation, and reduced grazing pressure are the principal factors responsible for this decline (Suárez 2010), but others not previously considered, such as the effects of fire, may also be involved. Despite the lack of knowledge on the consequences of fire on the shrub-steppes inhabited by Dupont's Lark, controlled burning has been proposed as a possible measure to prevent the growth of vegetation in the absence of grazing (Suárez 2010).

There are only some incidental data on burning in habitats occupied by the Dupont's Lark. Several small populations were found 15–20 years after wildfires in former pine forests that transformed them into shrubland, but the exact timing of the colonization is not known since no monitoring was carried out (Suárez 2010). On the other hand, a suitable semi-arid steppe in southeastern Spain (Almería Province) suffered a serious population decline after fires (Suárez 2010). In none of these examples were the changes in habitat variables described nor was the habitat use by the species monitored after the fire.

The aim of this study is to document the short-term effects of a summer wildfire that impacted an important Dupont's Lark population living in a habitat typical for the species, a low shrub-steppe dominated by *Genista pumila*. We analyse, for the first time, the changes in habitat structure caused by the fire in this type of plant formation, and its consequences for the abundance and habitat use of the Dupont's Lark. These results are important to assess the potential impact of fires on Dupont's Lark conservation and to evaluate the limitations of prescribed fires as a habitat management tool for conservation of this species.

STUDY AREA

The study area was located in the Layna páramos (Soria Province, Castilla y León Autonomous Community, Central Spain, 41°05'N, 1°50'W; Fig. 1) at 1,200 m.a.s.l., with a mean annual temperature of about 10 °C and a mean annual precipitation around 650 mm. This area exhibits an uncultivated flat relief, extending over 10,000 ha, that is dominated by low scrub comprised of *Genista pumila*, an species endemic to the Iberian Peninsula. It includes *Phlomis lychnitis*, *Lavandula latifolia*, *Thymus mastigophorus*, *T. vulgaris* and *T. mastichina*. The Layna páramos is one of the most important areas for the Dupont's Lark in Central Spain. This area is declared as a Special Protected Area (SPA) and Important Bird Area (IBA) and maintains one of the largest breeding populations of the Dupont's Lark, estimated at 200–250 pairs (Suárez 2010). Four-hundred and fifty hectares of the study area were accidentally burned in a 2009 summer fire.

METHODS

The Dupont's Lark breeding population was estimated using transects some years before fire (between 2004 and 2006) and the same transects were walked in the breeding season of 2010 and 2011, after the 2009 summer fire. Censuses were always carried out at approximately the same time of day (shortly before dawn) and on similar dates (April–May) and thus provide comparative data on population size and habitat use before and after the fire. Following the recommendations given by Garza et al. (2010), the singing males within a 500 m belt on each side of the transects were recorded via GPS. Transects were walked by three skilled ornithologists trained in this census method at the same research group (CPG, VG, JHJ). Three of the transects performed in the 2004–2006 period were affected by the wildfire (82% of its counting belt overlapped the burnt area, range 78–86%, 5,204 metres total length) and five were in the unburnt area (used here as a control, 13,100 metres total length; Fig. 1). Kilometric Abundance Index (KAI index hereafter; males/km) was calculated in each transect by dividing the total number of males recorded in each transect by the number of kilometres covered. The unburnt transects were located within three km from the burnt zone, and presented the same type of vegetation.

The effect of fire on habitat characteristics was studied by recording vegetation and soil variables at four circles of 10 m radius centred on equidistant points along each transect. Vegetation structure was estimated by recording the cover of the following substrates and vegetation: bare rock, pebble, bare soil, scrub and herbaceous plants. The cover of scrub and grasses was estimated at three height categories: <20 cm, between 20 cm and 40 cm, and >40 cm. The mean vegetation

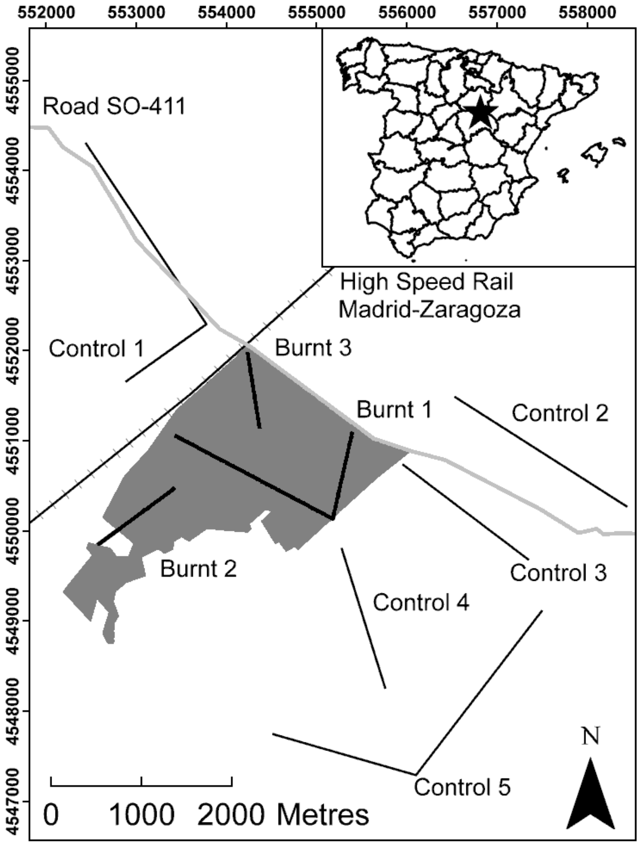

Fig. 1. Distribution of the transects and burnt zone (grey) in Layna. The inset shows the location of the study area (star) in Spain. UTM coordinates (zone 30) are shown.

height within each circle was visually estimated and the maximum height was recorded for the tallest plant. Cover was estimated following the method proposed by Prodon & Lebreton (1981). Vegetation variables were only measured in 2010 and 2011, after the fire. Circles were placed in the same location and variables recorded by the same observers (ESD, VN) in both years.

Maps of probability of Dupont's Lark presence in the area before and after the fire were developed using ArcGis Geostatistical Analyst (Johnston et al. 2001). Considering geo-referenced contacts from all visits to the transects since 2004, we recorded the use or lack thereof of each 0.1 km^2 pixel included within the area defined by the 500 m counting band on both sides of the transects. This was done separately for the periods before and after the fire, and the probability of use of pixels was interpolated in each period by ordinary kriging, a statistical procedure commonly used in geostatistics and ecology to interpolate values in unvisited locations from spatially distributed data, that takes into account the spatial autocorrelation of data (Fortin & Dale 2005). To map these results we divided the predicted probability into three categories by identifying two probability thresholds. First, the used pixel with the lowest predicted probability was identified and its probability value was considered as the threshold to separate unused and used areas. The second probability threshold was defined as the probability that maximizes the sum of the proportions of used pixels correctly classified (sensitivity) and unused pixels correctly classified (specificity; Franklin 2009), and discriminates the most suitable pixels. Probability thresholds were calculated separately for each map thus resulting in slightly different values.

Data analyses were conducted in R (R Development Core Team 2009) and results were expressed as means ± standard deviations. For transects that were counted more than once in the 2004–2006 period the mean number of males detected was used. The percentage of change in male number between the average in the pre-fire period and each year after the fire was calculated for each transect and these change percentages were compared between control and burnt areas using non-parametric Mann-Whitney U tests. A principal components analysis (PCA) was carried out on the habitat variables to identify main gradients of vegetation change due to fire. Data from the two years studied after the fire (2010 and 2011) were included in this analysis in order to assess the temporal change of vegetation in burnt and unburnt areas. The cover variables, which are expressed in percentages, were arcsine transformed. To determine the particular habitat variables that were affected by fire and how they changed between the first and the second spring after burning, we used mixed models (GLMM-S). A mixed model was fitted to each habitat variable including Zone type (control or burnt), Year (2010 and 2011) and Zone by Year interactions as fixed effects, and Transect and Vegetation Sampling Point nested in Transect as random effects. For this analysis we used the lme function in the R "nlme" package (Pinheiro et al. 2012).

RESULTS

A total of 55 and 18.5 males were recorded on average during the pre-fire years in the control and the subsequently burnt areas, respectively (Table 1). After the fire, in all burnt transects only three males were recorded in 2010 and one male in 2011, all of them located on transect margins, outside of the area of burnt vegetation. By contrast, the number of males in the control area remained closer to the pre-fire values although they also experienced some reduction (Table 1). There was no significant difference in mean KAI index (Kilometric Abundance Index) estimated in either zone before the fire in the 2004–2006 interval (4.20 in the control area and 3.28 in the burnt area; ANOVA: $F_{1,6} = 0.47$; $p = 0.52$), but the difference was significant after the fire both in 2010 and in 2011 (3.28 in the control area and 0.53 in the burnt area in 2010; ANOVA: $F_{1,6} = 11.07$, $p = 0.016$; 4.38 in the control area and 0.11 in the burnt area in 2011; ANOVA: $F_{1,6} = 6.1920$, $p = 0.047$).

Significant differences were found between the percentage of KAI change in the control transects (2010: -14.37%, 2011: 6.85%) and the transects overlapping the burnt area (2010: -85.54%; 2011: -97.4%; Mann-Whitney test; $z = -2.25$, $p = 0.024$; same test result for both years).

Maps produced from kriging interpolated values showed that habitat use in the control area essentially did not change during the study, while in the burnt zone the probabilities of the presence of a Dupont's Lark approaches zero (Fig. 2). Given that both maps were built using the presence data accumulated over several years the scarcity of used pixels in the burnt area reflects that the species avoided this area even in the second breeding season after the fire.

The first two axes of the PCA performed with habitat variables explained 61.87% of the variance (Table 2). The first factor was positively correlated with variables relating to lithological cover and negatively associated with those relating to plant cover, especially scrubs between 20–40 cm, scrubs > 40 cm and vegetation height, and it may thus be regarded as a gradient in the predominant substrate type, whether mineral or vegetational. The second axis was positively associated with bare

Table 1. Transect length, number of males recorded and KAI values (number of individuals per kilometre) in each transect in the pre-fire and post-fire periods. The percentage of change between pre-fire values and both post-fire breeding seasons is also shown. Totals and averages for control and burnt zones appear in the bottom lines.

Transect	Length (m)	Pre-fire (2004–2006)		Post-fire 1st year (2010)		Post-fire 2nd year (2011)		Change %	
		Males	KAI	Males	KAI	Males	KAI	1st year	2nd year
Control 1	3,391	10	2.95	8	2.36	8	2.36	-20.0	-20.0
Control 2	2,166	2.5	1.15	3	1.39	3	1.39	-25.0	-25.0
Control 3	1,827	9	4.93	8	4.38	16	8.76	-11.1	77.8
Control 4	2,082	13.5	6.48	8	3.84	11	5.28	-33.3	-8.3
Control 5	3,634	20	5.50	16	4.40	15	4.13	-20.0	-25.0
Burnt 1	3,002	13	4.33	2	0.67	1	0.33	-84.0	-92.3
Burnt 2	1,094	2.5	2.29	1	0.91	0	0	-50.0	-100.0
Burnt 3	928	3	3.23	0	0	0	0	-100.0	-100.0
Control zone	13,100	55	4.20	43	3.28	53	4.38	-14.37	6.85
Burnt zone	5,024	18.5	3.28	3	0.53	1	0.11	-81.54	-97.44

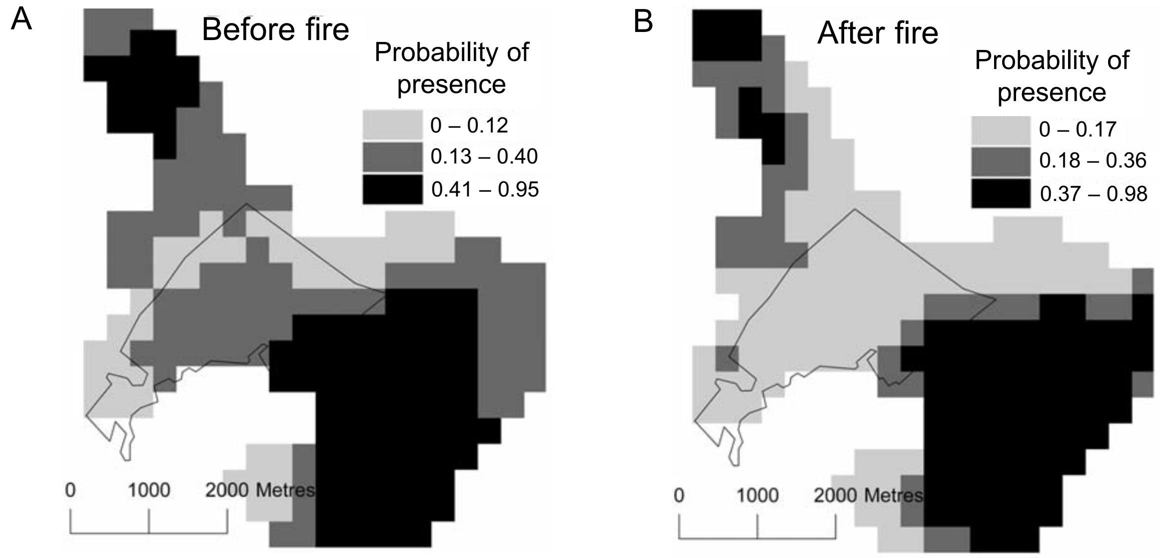

Fig. 2. Map of the probability of Dupont's Lark presence before (A)and after the fire (B) in the Layna páramos, interpolated by ordinary kriging. The polygon identifies the burnt zone. See text for definition of probability thresholds.

soil and negatively with herbaceous cover, and thus represents a gradient of grass development. Points in the control area show greater variability along Axis I while burnt points concentrate in positive coordinates, such that both zones are perfectly separated along this gradient (Fig. 3). Coordinates of control points remain practically unchanged between springs, while burnt points are displaced toward low values of Axis II in 2011, reflecting an increase in grass cover in the second spring.

The GLMM-s show that the zone effect was significant for all variables except scrub cover < 20 cm and grasses > 40 cm, and in most cases the interaction between zone and year was also significant (Table 2). The changes caused by fire were still apparent after two years in the burnt area. Scrubs > 20 cm were almost completely eliminated from this area and scrub cover remained low in the second spring, whereas grasses reached even greater cover two years after the fire (Table 2), thus passing from a vegetation dominated by low shrubs to one dominated by grassland (Fig. 3). On the contrary, bare rock, pebble and bare soil cover were higher in the burnt zone in the first spring after the fire. Changes in pebble and bare soil

Table 2. Mean ± SD of the habitat variables in unburnt and burnt areas in 2010 and 2011 and values of the loading factors of the PCA performed with habitat variables in both years. The significant effects from GLMM analysis performed with each variable are identified in the corresponding column with capital letters: Z — zone (burnt/unburnt); Y — year; I — Z × Y interaction, and asterisks. Bold font highlights the highest means when Zone effect was significant in GLMM. * — $p < 0.05$; ** — $p < 0.01$; *** — $p < 0.005$.

Variable	1st year Post-fire (2010)		2nd year Post-fire (2011)		GLMM	PCA	
	Unburnt area	Burnt area	Unburnt area	Burnt area	Effect Sig.	Axis I	Axis II
Bare rock	7.00 ± 7.32	**22.5 ± 14.38**	7.00 ± 7.32	**12.5 ± 6.57**	Z *	0.671***	0.139
Pebble	3.50 ± 4.00	**11.25 ± 5.27**	3.75 ± 3.93	**18.8 ± 3.77**	I ***	0.742***	-0.337**
Bare soil	23.00 ± 11.63	**45.41 ± 19.12**	**22.00 ± 10.56**	9.58 ± 3.34	I ***	0.333**	0.848***
Herb. < 20 cm	5.00 ± 3.97	5.84 ± 7.63	5.50 ± 4.56	**40 ± 5.64**	I ***	0.457***	-0.770***
Herb. 20–40 cm	**3.50 ± 5.15**	0	4.00 ± 5.28	**5.83 ± 3.58**	I ***	-0.328**	-0.674***
Herb. > 40 cm	0.50 ± 1.53	0	0.50 ± 1.54	0		-0.510***	-0.107
Scrub < 20 cm	27.75 ± 12.92	15 ± 6.03	26.5 ± 12.25	12.92 ± 4.50		-0.376***	0.192
Scrub 20–40 cm	**26.25 ± 14.58**	0	**27.00 ± 13.11**	0.37 ± 1.44	Z ***	-0.896***	0.127
Scrub > 40 cm	**3.75 ± 5.82**	0	**4.25 ± 5.68**	0	Z ***	-0.742***	-0.001
Maximum height	**32.25 ± 10.32**	7.92 ± 2.57	**33.00 ± 9.37**	15 ± 2.13	I ***	-0.920***	-0.045
Mean height	**19.00 ± 8.97**	5.83 ± 1.94	**20.50 ± 8.26**	9.16 ± 1.94	Z**,Y*	-0.895***	-0.105
Variance explained (%)						43.90	17.97

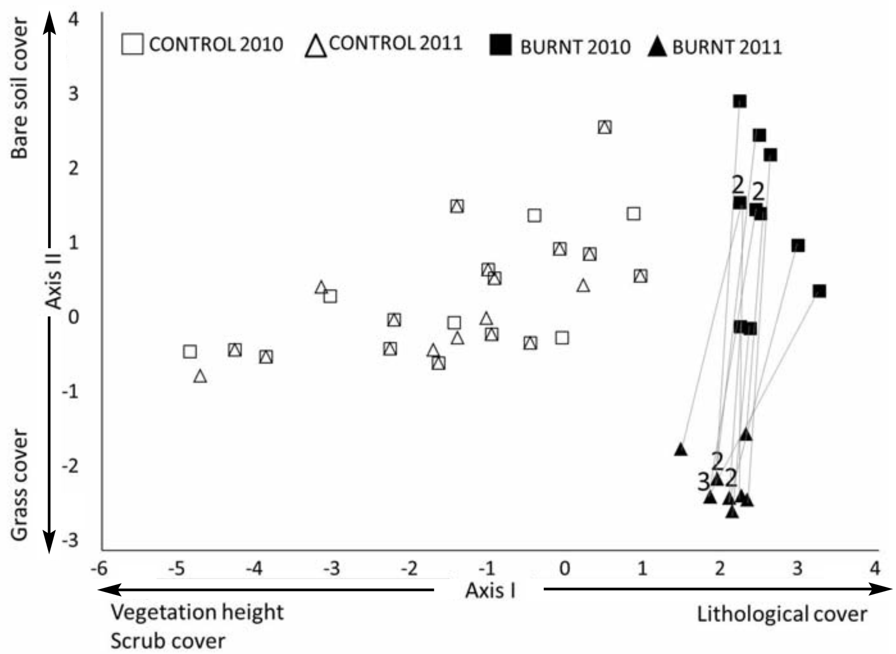

Fig. 3. Representation of vegetation sampling points in the first two axes resulting from principal components analysis. Figures close to the symbols represent the number of sampling points overlapping in some coordinates. Lines connect the positions in 2010 and 2011 of each sampling point in the burnt area. Interpretation of axes (arrows) is based on the correlations of habitat variables with each axis shown in Table 2.

cover between years were significant, but showed opposite effects. Bare soil, the most abundant substrate in 2010 in the burnt zone, decreased by up to 10% in 2011, while pebble cover increased (Table 2).

DISCUSSION

Our results show that the spring abundance of Dupont's Lark is drastically reduced in a burnt area even two years after the fire. The individuals that inhabited the burnt area do not seem to have been displaced by the fire into the neighbouring habitats, since the number of males in these areas also underwent some reduction. Thus, the fire provoked a decrease in the local population size in the studied area, but given the size of the Layna páramos area, we do not have data to assess whether the entire area population decreased or whether expelled males moved further away, occupying sub-optimal habitats, becoming diluted over a larger area. This outcome differs from effects found in forests, where fires create new habitats to be occupied by specialized and threatened Mediterranean species (Moreira et al. 2001, Pons & Bas 2005, Brotons et al. 2008, Clavero et al. 2011). The lack of studies carried out in low scrublands precludes comparison with previous works, but it is possible that fires of similar intensity could affect birds that strictly select low scrubland more seriously than those using high scrublands or forests. In forest fires, a proportion of trees usually survive and they may provide the necessary conditions for bird recruitment in burnt areas, as has been demonstrated with piled wood debris (Rost et al. 2012). However, in low scrublands, fires may burn all the above ground vegetation leaving no surviving shrubs, which may explain the marked effect registered in our study.

The analysis of habitat variables showed strong differences between control and burnt zones. No differences between years were detected in the control zone but in contrast the burnt area was dominated by bare ground in the first spring after the fire and by herbaceous cover in second spring. Scrub cover did not change very much between years, and thus the reduction of the amount of lithological cover in 2011 is due to the large increase in grasses in the lower height category. A similar increase in grass cover after fire has been found in other Mediterranean scrubs (Pons & Prodon 1996, Tárrega et al. 2001). However, pebble cover showed a contrary trend increasing in the second year after the fire, which could be due to soil erosion uncovering this substrate at some points (De Luis et al. 2003, Morgan 2005). After grasses, scrub under 20 cm was the most extensive vegetation category in the burnt zone, with a cover somewhat lower than that of the control area although not reaching significant differences between these areas. The dominant shrub species in the study area, *Genista pumila* (Garza et al. 2005), belongs to a genus that presents a high rate of germination after fire and is able to sprout after fire (Rivas et al. 2006, Reyes & Trabaud 2009), processes which could account for the relatively rapid recovery of low scrub cover in the first year. The recovery rate of low scrubs seems to have been largely attenuated in the second year and few plants appear to have grown into the next height category (20–40 cm).

Habitat selection studies in the same Layna páramos that include our study area (Seoane et al. 2006) found that Dupont's Larks select habitat patches with about 35% cover of medium-sized shrubs (25–50cm), and similar values have been obtained by Garza & Suárez (1990) in other Spanish localities. Radio-tracking of marked individuals also in Layna páramos showed that *Genista pumila*, which represents nearly 60% of scrubs between 20 cm and 40 cm in our study area, accounted for 90% of locations (Garza et al. 2005). The vegetation variables most profoundly affected by fire in our study area were precisely the cover of scrubs in height categories over 20 cm, which two years after the fire were nearly absent. Thus, given the habitat selection of this species, it may be expected that the burnt area would be unsuitable for this lark until the recovery of scrubs above 20 cm. This process could take at least four years if the results of studies on other Mediterranean shrubs were applicable to our study area (Tárrega et al. 2001, Calvo et al. 2002, Götzenberger et al. 2003), since *Genista pumila* has not been studied to date. Moreover, given that grasslands are considered a sub-optimal habitat (Seoane et al. 2006), the huge increase in grass cover in the second year after fire may contribute to declining habitat quality. Thus, although the effect of fire may be reversible, in the short term habitat quality is seriously reduced for several years, which could be due both to the disappearance of appropriate nesting substrate (Renwald 1977, Prodon & Pons 1993, Pons & Prodon 1996) and the shortage of food, which particularly affects insectivorous birds (Pons & Prodon 1996, Herrando et al. 2002).

Fires seem to have enabled the settlement of Dupont's Lark in previously forested areas, as has been observed in the localities of Tábara (Zamora Province, Northwest Spain) and Riodeva (Teruel Province, Central Spain), where the species was detected between 15 and 30 years after the fire in burnt areas transformed to shrub-steppes surrounded by pines. The population inhabiting a shrub-steppe dominated by *Stipa tenacissima* (La Chiripa, Murcia Province, Southeast Spain) is suspected to have benefited from a fire that occurred about 10 years before the species was detected for the first time (Suárez et al. 2009, Suárez 2010). This locality had been considered a potential area for the Dupont's Lark and thus had been visited by some researchers that did not find the species, which suggests that colonization occurred after the fire (Garza & Suárez 1990, Suárez 2010). By contrast, a wildfire led to the disappearance of a population established in a shrub-steppe at Las Amoladeras (Almería Province, Southeast Spain; Suárez 2010). Unfortunately no monitoring of vegetation or birds was carried out in these areas and thus uncertainty about the recovery process of both habitat and larks remains.

The dynamics of the bird community composition in burnt areas are in most cases closely linked to the time elapsed after the fire (Jacquet & Prodon 2009), since wildfires maintain the availability of open patches (Lloret et al. 2002, Pons & Bas 2005, Clavero et al. 2011), many of which disappear in the middle or long-term following fire (Preiss et al. 1997, Lloret et al. 2002, Jacquet & Prodon 2009, Pons & Clavero 2010, Pons et al. 2012). Long-term effects of fires may be helpful for maintaining landscape heterogeneity and promoting dwarf shrub regeneration, which today is very dense due to the socio-economic drift that has promoted the abandonment of grazing and marginal agricultural areas and the subsequent growth of scrubs over optimal height for the species (Preiss et al. 1997, Clavero et al. 2011). Several studies indicate that fires have positive consequences for scrub bird species in Mediterranean pine forests, even in the short-term, and some passerines such as Sardinian and Dartford Warblers (*Sylvia melanocephala* and *Sylvia undata*) colonized burnt areas as early as the second year after fire (Pons et al. 2012). Another threatened Mediterranean species favoured by the creation of openings by fires in forested areas are Ortolan Bunting *Emberiza hortulana* (Menz et al. 2009), Black Wheatear *Oenanthe leucura* (Real 2000) and Black-eared Wheatear *Oenanthe hispanica* (Pons &

Prodon 1996). The effect of fire on open habitats has been studied mainly in North American prairies, where burning and grazing are fundamental processes for avoiding encroachment (Fuhlendorf et al. 2006, Coppedge et al. 2008, Murphy 2008, Powell 2008). Most studies have found differing responses of bird species, as a function of specific life-history traits and habitat preferences, and most agree that, although some species are negatively affected in the first breeding season, their abundance returned to pre-fire levels within 2–3 years (Grant et al. 2010, Roberts et al. 2012). It has been suggested that most species of breeding grassland birds are generally well-adapted to fire that recurs every 4 to 6 years (Murphy 2008, Powell 2008). One important difference between these ecosystems and our study area is that they are dominated by grasslands, which recover faster than scrubs as our results for the second spring after fire have shown. However, as stated previously, in Spain a continuous grass cover is sub-optimal for the species that also selects for some percentage of bare ground (Seoane et al. 2006). In this country just a few populations of Dupont's Lark inhabit alpha grass *Stipa tenacissima* steppes while most North African populations (90%) are located in areas of this vegetation type (García et al. 2008, Suárez 2010). The cover of alpha grass in Mediterranean areas has been found to grow to pre-fire levels within 2–3 years (Martínez-Sánchez et al. 2007); thus, it is possible that Dupont's Lark populations living in alpha grass steppes could recover after fire faster than those inhabiting scrublands, but information on this possibility is lacking.

Our results have shown that the consequences of intense wildfires affecting steppe scrublands may be more damaging and in the case of the Dupont's Lark may lead to the disappearance of the species in the short-term from burnt areas. The strict habitat selection of Dupont's Lark likely plays an important role in this process and thus, the recovery of vegetation characteristics required by the lark may take longer than for other more eclectic species. Thus, even in the hypothetical situation that the shrub recovery leads to better habitat quality than before fire on a broader temporal scale, it is unlikely that this will compensate for the loss of part of the population and their contribution to recruitment during the transitional years.

Dupont's Lark conservation, as well as many other open-country species, depends on maintaining large areas in early stages of plant succession. Habitat management by grazing, perhaps involving limited prescribed fire may help to conserve it, particularly now that the former is in sharp decline. However, our results show that uncontrolled fires affecting optimal habitat have negative consequences for the species that may last several years. Thus, the use of prescribed fires as a habitat management tool for this species requires more research and careful evaluation, and we believe that at the present level of knowledge it should be restricted to tall dense scrubland currently unsuitable for the species. Potential candidate areas could be flat unoccupied parcels close to Dupont's Lark populations, where prescribed controlled fires could be used to create new open-habitats, increase the lark's geographical spread and promote connectivity between population nuclei. In areas where the Dupont's Lark is present we recommend the use of alternative vegetation management strategies, such as the return of extensive grazing, while limiting new land uses such as wind farms, reforestation or ploughing for cultivation. Future research efforts examining long-term effects of fire in Dupont's Lark populations could yield more precise recommendations on the use of prescribed fire as a conservation tool for this species and also on restoration measures for the areas affected by wildfires.

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REFERENCES

BirdLife International 2004. Birds in Europe: population estimates, trends and conservation status. BirdLife Conservation Series No. 12. BirdLife International, Cambridge, UK.

Bouwman H., Hoffman R. 2007. The effects of fire on grassland bird communities of Barberspan, North-West Province, South Africa. Ostrich 78: 591–608.

Brotons L., Herrando S., Pons P. 2008. Wildfires and the expansion of threatened farmland birds: the ortolan bunting *Emberiza hortulana* in Mediterranean landscapes. J. Appl. Ecol. 45: 1059–1066.

Calvo L., Tárrega R., De Luis E. 2002. The dynamics of Mediterranean shrubs species over 12 years following perturbations. Plant Ecol. 160: 25–42.

Clavero M., Brotons L., Herrando S. 2011. Bird community specialization, bird conservation and disturbance: the role of wildfires. J. Anim. Ecol. 80: 128–136.
Coppegde B. R., Fuhlendorf S. D., Harrell W. C., Engle D. M. 2008. Avian community response to vegetation and structural features in grasslands managed with fire and grazing. Biol. Conserv. 141: 1196–1203.
Dechant J. A., Sondreal M. L., Johnson D. H., Igl L. D., Goldade C. M., Parkin B. D., Euliss B. R. 2003. Effects of management practices on grassland birds: Lark Sparrow. Northern Prairie Wildlife Research Center, Jamestown. Nebraska, USA.
De Luis M., González-Hidalgo J. C., Raventós J. 2003. Effects of fire and torrential rainfall on erosion in a Mediterranean gorse community. Land Degrad. Dev. 14: 203–213.
Fortin M. J., Dale M. R. 2005. Spatial Analysis. A Guide for Ecologists. Cambridge University Press, Cambridge, UK.
Franklin J. 2009. Mapping species distributions. Spatial inference and prediction. Cambridge University Press, Cambridge, UK.
Fuhlendorf S. D., Harrell W. C., Engle D. M., Hamilton R. G., Davis C. A., Leslie D. M. 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecol. Appl. 16: 1706–1716.
García J. T., Suárez F., Garza V., Justribó J. H., Oñate J. J., Hervás I., Calero M., García de la Morena E. 2008. Assessing the distribution, hábitat, and population size of the threatened Dupont's lark *Chersophilus duponti* in Morocco: lessons for conservation. Oryx 42: 592–599.
Götzenberger L., Ohl C., Hensen I., Sánchez P. 2003. Postfire regeneration of a thermomediterranean shrubland area in south-eastern Spain. An. Biol. 25: 21–28.
Garza V., Suárez F. 1990. [Distribution, population and habitat selection of Dupont's Lark (*Chersophilus duponti*) in the Iberian peninsula]. Ardeola 37: 3–12.
Garza V., Suárez F., Carriles E. 2010. [The actual census: design and sampling methods and populations estimates]. In: Suárez F. (ed.). [Dupont's Lark *Chersophilus duponti*]. Dirección General para la Biodiversidad. Ministerio de Medio Ambiente y Medio Rural y Marino, Madrid, Spain, pp. 175–194.
Garza V., Suárez F., Herranz J., Traba J., García de la Morena E. L., Morales M. B., González R., Castañeda M. 2005. Home range, territoriality and habitat selection by the Dupont's Lark *Chersophilus duponti* during the breeding and post-breeding periods. Ardeola 52: 133–146.
Grant T. A., Madden E. M., Shaffer T. L., Dockens J. S. 2010. Effects of prescribed fire on vegetation and passerine birds in northern mixed-grass prairie. J. Wildl. Manage. 74: 1841–1851.
Herrando S., Brotons L. 2002. Forest bird diversity in Mediterranean areas affected by wildfires: a multiscale approach. Ecography 25: 161–172.
Herrando S., Brotons L., Del Amo R., Llacuna S. 2002. Bird community succession after fire in a dry Mediterranean shrubland. Ardea 90: 303–310.
Jacquet K., Prodon R. 2009. Measuring the post-fire resilience of a bird–vegetation system: a 28-year study in a Mediterranean oak woodland. Oecologia 161: 801–811.
Johnston K., Ver Hoef J. M., Krivoruchko K., Lucas N. 2001. Using ArcGis Geostatistical Analyst. GIS by ESRI. ESRI Press, California, USA.
Lloret F., Calvo E., Pons X., Díaz-Delgado R. 2002. Wildfires and landscape patterns in the Eastern Iberian Peninsula. Landscape Ecol. 17: 745–759.
Martínez-Sánchez J. J., Herranz J. J., Guerra J., Trabaud L. 1997. Influence of fire on plant regeneration in a *Stipa tenacissima* L. community in the Sierra Larga mountain range (SE Spain). Israel J. Plant Sci. 45: 309–316.
Menz M. H., Brotons L., Arlettaz R. 2009. Habitat selection by Ortolan Buntings *Emberiza hortulana* in post-fire succession in Catalonia: implications for the conservation of farmland populations. Ibis 151: 752–761.
Moreira F., Delgado A., Ferreia S., Borralho R., Oliveira N., Inácio M., Silva J. A., Rego F. 2003. Effects of prescribed fire on vegetation structure and breeding birds in young *Pinus pinaster* stands of northern Portugal. Forest Ecol. Manage. 184: 225–237.
Moreira F., Rego F. C., Ferreira P. G. 2001. Landscape changes and breeding bird assemblages in northwestern Portugal: the role of fire. Landscape Ecol. 16: 175–187.
Morgan R. 2005. Soil erosion and conservation. National Soil Resources Institute. Cranfield University, Blackwell Science, Oxford, UK.
Murphy R. 2008. Fire is for the birds in northern mixed-grass prairie. Fire Sci. Brief. 9: 1–6.
Nappi A., Drapeau P. 2009. Reproductive success of the black-backed woodpecker (*Picoides arcticus*) in burned boreal forests: Are burns source habitats? Biol. Conserv. 142: 1381–1391.
Pinheiro J., Bates D., DebRoy S., Sarkar D., R Development Core Team. 2012. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3. 1–105.
Pons P., Bas J. M. 2005. Open-habitat birds in recently burned areas: the role of the fire extent and species habitat breadth. Ardeola 52: 119–131.
Pons P., Clavero M. 2010. Bird responses to fire severity and time since fire in managed mountain rangelands. Anim. Conserv. 13: 294–305.
Pons P., Clavero M., Bas J. M., Prodon R. 2012. Time-window of occurrence and vegetation cover preferences of Dartford and Sardinian Warblers after fire. J. Ornithol. 153: 921–930.
Pons P., Prodon R. 1996. Short-term temporal patterns in a Mediterranean shrubland bird community after wildfire. Acta Oecol. 17: 29–41.
Powell A. 2008. Responses breeding birds in tall grass prairie to fire and cattle grazing. J. Field. Ornithol. 79: 41–52.
Preiss E., Martin J. L., Debussche M. 1997. Rural depopulation and recent landscape changes in a Mediterranean region: consequences to the breeding avifauna. Landscape Ecol. 12: 51–61.
Prodon R., Lebreton J. D. 1981. Breeding avifauna of a Mediterranean sucession: the holm oak and cork oak series Pyrenees. I. Analysis and modelling of the structure gradient. Oikos 37: 21–38.
Prodon R., Pons P. 1993. Postfire bird studies: methods, questions and perspectives. In: Trabaud L., Prodon R. (eds). Fire in Mediterranean Ecosystems. Ecosystems Research Report 5. Commission of the European Communities, Brussels, Belgium, pp. 332–343.
R Development Core Team. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org.
Real J. 2000. [Wildfires could favour Black Wheatear (*Oenanthe leucura*) recolonisation]. Ardeola 47: 93–96.
Renwald J. D. 1977. Effect of fire on Lark Sparrow nesting densities. J. Range Manage. 30: 283–285.
Reyes O., Trabaud L. 2009. Germination behaviour of 14 Mediterranean species in relation to fire factors: smoke and heat. Plant Ecol. 202: 113–121.
Rivas M., Reyes O., Casal M. 2006. Influence of heat and smoke on the germination of six leguminous shrubby species. Int. J. Wildland Fire 15: 73–80.
Roberts A. J., Boal C. W., Wester D. B., Ridenout-Hanzak S., Whitlaw H. A. 2012. Grassland bird community response to large wildfires. Wilson J. Ornithol. 124: 24–30.

Rost J., Clavero M., Bas J. M., Pons P. 2010. Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests. Forest Ecol. Manag. 260: 79–86.
Seoane J., Justribó J. H., García F., Retamar J., Rabadán C., Atienza J. C. 2006. Habitat-suitability modelling to assess the effects of land-use changes on Dupont's Lark *Chersophilus duponti*: A case study in the Layna Important Bird Area. Biol. Conserv. 128: 241–252.
Suárez F. (ed). 2010. [Dupont's Lark *Chersophilus duponti*]. Dirección General para la Biodiversidad. Ministerio de Medio Ambiente y Medio Rural y Marino, Madrid, Spain.
Suárez F., Hervás I., Herranz J. 2009. [Larks in mainland Spain]. Dirección General para la Biodiversidad. Ministerio de Medio Ambiente y Medio Rural y Marino, Madrid, Spain.
Tárrega R., Luis-Calabuig E., Valbuena L. 2001. Eleven years of recovery dynamic after experimental burning and cutting in two *Cistus* communities. Acta Oecol. 22: 277–283.
Tella J. L., Vögelli M., Serrano D., Carrete M. 2005. Status of the threatened Dupont's Lark in Spain: overestimation, decline and extinction of local populations. Oryx 39: 1–5.

STRESZCZENIE

[Krótkoterminowe efekty pożarów na występowanie skowrończyka sierpodziobego w centralnej Hiszpanii]

Pożary mogą pozytywnie wpływać na ochronę niektórych zagrożonych gatunków awifauny śródziemnomorskiej, ponieważ powstałe w ich wyniku siedliska otwarte na terenach leśnych są często zajmowane przez takie gatunki, a dodatkowo w związku z odradzaniem się roślinności po pożarze poprawie może ulec jakość siedliska.

W przypadku skowrończyka sierpodziobego, jednego z najbardziej zagrożonych wyginięciem wróblaków zasiedlających tereny stepowe w Europie, kontrolowane pożary są czasem proponowane jako środek służący ochronie czynnej tego gatunku. Brak jest jednak dokładnych analiz co do skuteczności wpływu takich zabiegów na wzrost liczebności populacji tego gatunku.

Prace prowadzono w centralnej Hiszpanii, gdzie znajduje się jedna z największych populacji lęgowych skowrończyka sierpodziobego obejmująca ok. 200–250 par. Teren ten porośnięty jest niskimi krzewami zdominowanymi przez janowiec *Genista pumila*. W 2009 r. 450 ha tego terenu uległo wypaleniu w wyniku przypadkowego pożaru. Liczenia ptaków, które przeprowadzono w sezonach 2004–2006, a następnie 2010–2011 umożliwiły prześledzenie zmian w liczebności populacji lęgowej skowrończyka, które nastąpiły po pożarze. Badania prowadzono na trzech transektach przebiegających przez teren pożarzyska oraz pięciu zlokalizowanych w najbliższej okolicy, traktowanych jako kontrola (Fig. 1). Dla wszystkich ośmiu transektów zbierano dane o liczbie i rozmieszczeniu śpiewających samców. Na podstawie miejsc stwierdzenia ptaków w terenie stworzono następnie mapy rozkładu prawdopodobieństwa występowania badanego gatunku. Na transektach kontrolnych, jak i tych zlokalizowanych na pożarzyskach, opisano także strukturę siedliska.

Na terenach po pożarze stwierdzono spadek liczebności skowrończyków o 85–100% w porównaniu do okresu przed pożarem, natomiast na terenach kontrolnych ten spadek był znacznie niższy i wynosił 7–15% (Tab. 1). Prawdopodobieństwo występowania na terenach nie objętych pożarem nie uległo zmianie, natomiast na obszarze pożarzyska spadło prawie do zera (Fig. 2). Stwierdzono istotne zmiany w siedlisku na terenie pożarzyska w porównaniu z obszarami kontrolnymi (Tab. 2, Fig. 3). Na badanym terenie po pożarze znikły krzewy, które dopiero w kolejnych latach powoli się regenerowały, stwierdzono także silny wzrost pokrycia terenu przez trawy, co wpływa na pogorszenie jakości siedliska tego gatunku.

Autorzy wskazują, że pożary są niewskazane w miejscach zasiedlanych przez skowrończyka, gdyż krótkoterminowo mogą doprowadzić do lokalnych ekstynkcji. Jednocześnie sugerują, że kontrolowane wypalanie można prowadzić na okolicznych terenach pokrytych wysokimi i gęstymi zaroślami nieodpowiednimi dla tego gatunku, tworząc nowe tereny otwarte, które mogłyby być potem sukcesywnie kolonizowane przez skowrończyki, jak i inne zagrożone gatunki preferujące tereny otwarte.